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Soil carbon pools and fluxes after land conversion in a semiarid shrub-steppe ecosystem

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Abstract Worldwide soil carbon (C) losses associated with agricultural expansion and intensification have contributed significantly to increased atmospheric CO_2. Soil disturbances resulting from land use changes were shown to modify the turnover of C and the formation of soil organic matter. A native semiarid shrub-steppe ecosystem recently converted into an irrigated agricultural development in the Columbia Basin of Washington State was evaluated for several abiotic indicators that might signal changes in an ecosystem during the initial stages of conversion and disturbance. Soil samples were collected in March of 2003 and 2004 from nine sites that included native shrub-steppe and agricultural fields converted in 2001 and 2002. Disturbance from conversion to irrigated crop production influenced total organic C and nitrogen (N) storage, C and N mineralization, and C turnover. Cultivated fields had greater concentrations of total organic C and N and higher cumulative C and N mineralization than native sites after 3 years of cultivation. Soil organic C was divided into three pools: an active pool (C_a) consisting of labile C (simple sugars, organic acids, the microbial biomass, and metabolic compounds of incorporated plant residues) with a mean residence time of days, an intermediate or slow pool (C_s) consisting of structural plant residues and physically stabilized C, and a resistant fraction (C_r) consisting of lignin and chemically stabilized C. Extended laboratory incubations of soil with measurements of CO_2 were used to differentiate the size and turnover of the C_a and C_s functional C pools. The active pools were determined to be 4.5 and 6.5% and slow pools averaged 44 and 47% of the total C in native and cultivated fields, respectively. Cultivation, crop residue incorporation, and dairy manure compost amendments contributed to the increase in total soil C.

Keywords C cycling · Carbon pools · C turnover · C mineralization · Arid shrub-steppe

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Introduction

Worldwide soil carbon (C) losses associated with agricultural expansion and intensification during the past 150 years have significantly contributed to increased atmospheric CO_2 (Houghton et al. 1983; Post et al. 1990; Flach et al. 1997). Soil disturbances resulting from land use changes were shown to modify the turnover of C and the formation of soil organic matter (SOM) (Huggins et al. 1998; Collins et al. 2000; Dinesh et al. 2003). The conversion of native ecosystems to agricultural developments usually results in a loss of SOM. Such losses are well documented for the Great Plains and Corn Belt (Paustian et al. 1997), with few examples in the Pacific Northwest (Rasmussen et al. 1980; Rasmussen and Collins 1991).

Conversion of the native shrub-steppe in the semiarid region of the Columbia Basin in Eastern Washington, USA to high input production agriculture has increased with the availability of water. In 1942, the Columbia Basin Irrigation Project began with the construction of Grand Coulee Dam on the Columbia River (Bureau of Reclamation 2005, US Department of the Interior 2005). Irrigation water was first delivered to agricultural fields in the early 1950s, irrigating approximately 26,710 ha (Hubbard 1996). As water availability increased, land conversions expanded to over 271,500 ha of irrigated land in 2005 (Bureau of Reclamation 2005, US Department of the Interior 2005).

Early land conversions in the Basin consisted of moderate forms of land clearing and land leveling to facilitate gravity systems such as furrow (ditch) irrigation. Land leveling is a common method of land preparation for irrigation where a uniform slope is established to provide an even distribution of irrigation water, to conserve soil, and to ensure the stability of cropping (Whitney et al. 1950; Brye et al. 2003). Irrigation improvements to more efficient pressurized systems such as center pivot have since dominated (Evans et al. 2000) and land leveling operations have intensified. Also, with the expansion of center pivot systems, land clearing and leveling of marginal lands has increased. Marginal lands were those once considered unsuitable for irrigation because of sandy soil and varied topography (Evans et al. 2000).

Land leveling can result in loss of topsoil, exposure of subsoil, reduced organic carbon (OC), and a decrease in microbial activity (Brye et al. 2003). Loss of topsoil from disturbance could have negative impacts on ecosystem function because soil microbial biomass carbon and microbial biomass nitrogen from native surface soils (0–5 cm) in the shrub-steppe were found to have two to four times higher values than subsoil (5–15 cm) (Bolton et al. 1993).

While land leveling may reduce and redistribute surface soil organic carbon (SOC), the use of irrigation can increase plant production by producing greater crop yields that contribute to increased SOC from higher inputs of plant residues and root systems (Lueking and Schepers 1985; Entry et al. 2002). Irrigation may also increase the rate of plant residue decomposition because of increased soil moisture (Kumar and Goh 2000). Native areas in the semiarid Sandhills of Nebraska with low values of total C and N showed increased concentrations 15 years after the development of irrigation (Lueking and Schepers 1985), while soils with higher initial concentrations of C and N decreased only slightly. Entry et al. (2002) found that irrigated soils in southern Idaho under different agricultural practices (pasture and conservation tillage) exhibited significant increases in SOC over the native sagebrush ecosystem. The increase in SOC in irrigated agricultural sites is contradictory to conventional rainfed agricultural fields when compared with native sites (Paustian et al. 1997; Entry et al. 2002). Semiarid shrub-steppe ecosystems differ from other native systems (forests, permanent grasslands, or native prairies) because of the relatively small amount of annual precipitation and the lower levels of C inputs and storage in soil (Bolton et al. 1990; Entry et al. 2002; Smith et al. 2002). Sala et al. (1988) reported a correlation between annual precipitation and net primary productivity (NPP) in grasslands of the central United States showing soils receiving limited precipitation (<200 mm) generate minimal aboveground biomass. Rickard and Vaughan (1988) reported average above ground litter biomass as 191 g m^{-2} in a relatively undisturbed sagebrush–bunchgrass plant community in Eastern Washington compared to an annual above ground NPP of 236 g m^{-2} $year^{-1}$ in ungrazed grasslands of Western North America (Sims and Singh 1978) and 153.6 g m^{-2} $year^{-1}$ in older landscapes of the Alaskan tundra (Hobbie and Gough 2004).

Areas of the Columbia Basin are still experiencing agricultural growth with conversion of the remnant native shrub-steppe to irrigated organic high value vegetable production. There is limited knowledge of the effects of soil disturbance on C cycling resulting from land conversion of semiarid native systems in the Pacific Northwest, especially in the early stages of transition. The objective of this study was to document changes that occur in SOC during the transition of a semiarid native shrub-steppe to one under irrigated crop production.

Materials and methods

The study area was located in Grant County, Washington in a native shrub-steppe ecosystem that had been converted to a series of organic, irrigated agricultural fields. The shrub-steppe is a portion of the semiarid treeless ecotone in the Western United States that stretches from British Columbia, Canada, to Mexico. Native vegetation in the area includes big sagebrush (*Artemisia tridentate* Nutt), antelope bitterbrush (*Purshia tridentate* Pursh DC), rabbit brush [*Chrysothamnus viscidiflorus* (Hook.) Nutt], and other species of sagebrush. Other vegetation includes grasses such as bluebunch wheatgrass [*Pseudoregneria spicata* (Pursh) A Löve], Idaho fescue (*Festuca idahoensis* Elmer), needlegrass [*Achnatherum nelsonii* (Scribn.) Barkworth], and Sandberg's bluegrass (*Poa secunda* J. Presl) (Rickard 1988; Rogers and Rickard 1988; Wooten 2002). Annual rainfall ranged from 79 to 185 mm during the years of this study with most of the precipitation occurring in the winter months with a mean annual temperature (MAT) of 10.5°C (Fig. 1). The soil was a fine grain eolian sand and was classified as a mixed mesic Xerollic Cambothids (Adkins series). Soil horizons are brown, very fine sandy loams to a depth of 1.5 meters.

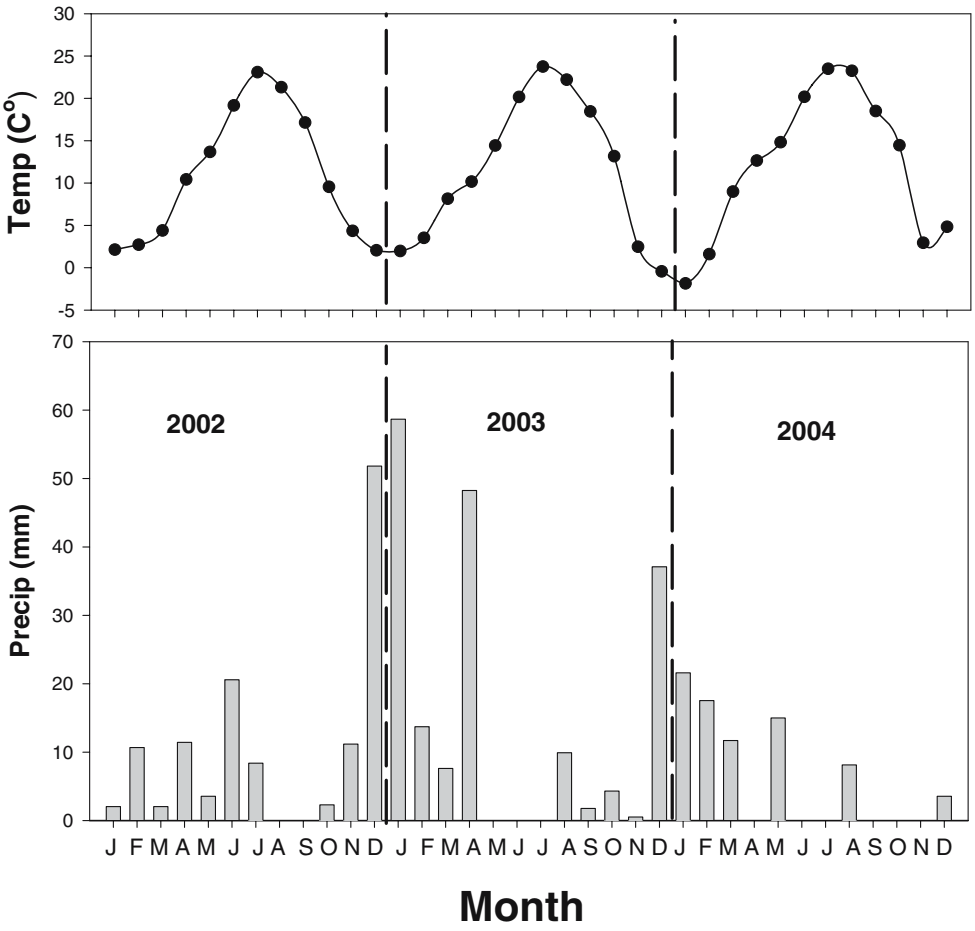

Fig. 1 Average monthly temperatures in degree Celsius and precipitation for Royal City, Grant, County, WA, USA for years 2002–2004. Data provided courtesy of Washington State University, Center for Precision Agricultural Systems, Prosser, WA, USA

The soil has weak structure containing little or none of the binding agents, clay or organic matter.

A chronosequence of nine sites in relatively close proximity south of Royal City, Washington (latitude: 46.9° N, longitude: 119.6° W, and elevation: 329 m) were sampled in the spring of 2003 and 2004. In 2003, the sites included three native sites (NS series) and six cultivated fields; three with 1 year of cultivation (CA series) and two with 2 years of cultivation (CB series). The sites were resampled in 2004 (Table 1).

Before cultivation, land management consisted of cattle grazing on open rangeland. In 2001, conversion from the shrub-steppe began with the removal of native vegetation by burning and leveling the land to facilitate the installation of a center pivot irrigation system and the subsequent planting of vegetable crops. The source of irrigation water was from the regional underground aquifer. After clearing, cultivated sites were tilled with the incorporation of residues and amendment of organic composted dairy manure. In 2002 and 2003, the compost was added before planting at a rate of 11.2 and 22.4 T ha^{-1}, respectively. Cultivated fields were managed in a 3-year rotation of sweet corn/peas/sweet corn. Crop yields were obtained from records kept by the grower/cooperator.

Soil samples were collected in March of 2003 and 2004 before field operations started, which corresponded to the time of year when maximum primary productivity occurs in the shrub-steppe (Rickard 1988). The interaction of increasing temperatures and soil moisture content during the spring season produces the greatest microbial activity and respiration in semiarid shrub-steppe soils (Wildung and Garland 1988). Soil samples were collected at 18-m intervals along two 100-m transects across each field for a total of 90 samples. Soil was sampled to a depth of 20 cm with a 2.5-cm soil probe with 15 composite cores taken in a 1-m^2 area around each location. Soils were transported on ice and stored at 4°C until analyzed.

Soil water-holding capacity was determined using a volumetric soil–water method described by Hook and Burke (2000). Briefly, air-dried sieved (2 mm) soil was packed lightly into 50-cm^3 graduated cylinders and 5 ml of distilled water was slowly added. The cylinder was covered with perforated parafilm (American National Can™, Greenwich, CT, USA) and allowed to equilibrate. After 24 h, soil volume and water content of the wetted front was determined. Bulk density of each sample was determined by dividing the weight of the dry soil by the volume within the cylinders. Water-holding field capacity was calculated

Table 1 Site history listing year of conversion from native shrub-steppe, vegetation and crop rotations for native and cultivated sites from 2001 through 2003

Site	Year converted	Hectares	2001 vegetation 2001 season	2002 vegetation 2002 season	2003 vegetation 2003 season
NS-1		–	Native veg.[a]	Native veg.	Native veg.
NS-2		–	Native veg.	Native veg.	Native veg.
NS-3		–	Native veg.	Native veg.	Native veg.
C A-1	2002	37.6	Native veg.	Sweet corn	Peas
C A-2	2002	47.8	Native veg.	Sweet corn	Peas
C A-3	2002	44.5	Native veg.	Sweet corn	Peas
C B-1	2001	25.9	Sweet corn	Peas	Sweet corn
C B-2	2001	48.6	Sweet corn	Peas	Sweet corn

NS Native shrub-steppe sites, *CA*=1 and 2 years of cultivation, and *CB* 2 and 3 years of cultivation at time of sampling
[a]Native veg. includes big sagebrush (Artemisia tridentate Nutt), antelope bitterbrush (*Purshia tridentate* Pursh DC), rabbit brush [*Chrysothamnus viscidiflorus* (Hook.) Nutt], and grasses such as bluebunch, wheatgrass [*Pseudoregneria spicata* (Pursh) A Löve], Idaho fescue (*Festuca idahoensis* Elmer), needlegrass [*Achnatherum nelsonii* (Scribn.) Barkworth], and Sandberg's bluegrass (*Poa secunda* J. Presl)

from the equation $W_{FC}=\theta_A/\theta_S$ where W_{FC} is the field capacity, θ_A it the weight of water added, and θ_S is the weight of soil wetted by the addition of water. Soil water content at the time of sampling was determined on subsamples by oven drying at 105°C for 24 h (Gardner 1986). All results are reported on an oven dry soil basis.

Soil pH was determined using a 2:1 water method (Robertson et al. 1999). Ten grams of soil was mixed with 20 ml of distilled water in 50-ml centrifuge tubes, capped, and shaken for 5 min at low speed. After shaking, caps were removed and the soil slurry was allowed to equilibrate for 30 min. Soil pH was determined using a Fisher Scientific Accumet Basic 15 pH meter (Hampton, NH, USA).

Total soil organic C (TOC) and N (TN) were analyzed by dry combustion on a LECO, CNS-2000 Elemental Analyzer, St. Joseph, MI, USA. Soil samples were prepared by sieving subsamples through a 2-mm screen with hand removal of recognizable plant materials. Carbonates were removed by mixing 100 ml of 250 mM HCl acid with 20 g soil and shaking on a reciprocal shaker for 1 h. Soil was washed three times with deionized water and centrifuged to remove excess chloride ions (Collins et al. 2000). Soil was then dried at 55°C and weighed for analysis.

C mineralization was measured using the static incubation method (Zibilske 1994). Twenty-five grams of soil samples were adjusted to 70% field capacity in 100-ml containers equipped with rubber septa and incubated at 25°C for an average of 175 days. Headspace CO_2 was measured weekly by direct injection of gas samples into an infrared gas analyzer (Analytical Development, type 225-MK3, Hertfordshire, England, UK). After analysis, samples were returned to ambient air by degassing with compressed air and adjusted for any moisture loss.

SOC has generally been divided into three pools: an active pool (C_a) consisting of labile C (simple sugars, organic acids, microbial biomass, and metabolic compounds of incorporated plant residues) with a mean residence time (MRT) of days; an intermediate or slow pool (C_s) consisting of structural plant residues and physically stabilized C with an MRT of 25–50 years; and a resistant fraction (C_r) consisting of lignin and chemically stabilized C with an MRT of 1,000–1,500 years (Buyanovsky et al. 1994).

The use of extended laboratory incubations of soil with measurements of CO_2 was widely used to differentiate the C_a and C_s functional C pools in residues (Collins et al. 1990) and soil (Motavalli et al. 1994; Paul et al. 1999, 2001b; Collins et al. 2000). This method can be considered a biological fractionation of organic matter whereby the labile (active) fractions (C_a) of SOM are rapidly mineralized by soil microorganisms and subsequent soil C (C_s) fractions are more slowly mineralized. The CO_2 evolved during C mineralization was used to determine the size and kinetics of the functional C pools of soil for each site (Paul et al. 1999). The size and turnover rates of each pool were estimated by curve fitting the CO_2 evolved per unit of time (C_t) using a three-component first-order model:

$$C_t = C_a e^{-k_a t} + C_s e^{-k_s t} + C_r e^{-k_r t}$$

where C_a and k_a = active pool, C_s and k_s = slow pool, and C_r and k_r = resistant pool. Three parameters, C_a, k_a, and k_s, were estimated using the nonlinear regression model (NonLIN) of Systat (Evanston, IL, USA). Because the residue of acid hydrolysis typically C-dates greater than 500 years, it was assumed that negligible amounts of the CO_2 evolved during the extended incubation were derived from the C_r pool (Paul et al. 1997). This assumption made it possible to analyze the CO_2 data as the sum of two first-order rate reactions. The slow pool C_s pool was defined as $C_s=C_t-C_a-C_r$. The model was based on the assumption of first-order kinetics, i.e., where the rate of C mineralization is proportional to the amount of C in the organic matter pool. When integrated over time this produces an expo-

nential decay curve. MRT was reported as the reciprocal of the decomposition rate constant (k^{-1}) derived from the first-order rate reaction. The MRT derived from laboratory incubation at 25°C was scaled to the MAT by assuming a Q_{10} of 2, $\left(2^{(25-t)/10}\right)$ where t is the MAT of 10.5°C (Mummey et al. 1994). Acid hydrolysis determined the size of the resistant C pool (C_r). The acid resistant organic fraction was determined by digesting 1 g of soil in 6 N of HCl for 18 h. Digested samples were washed three times with deionized water to remove excess Cl^-, dried at 55°C, and ground to pass a 180-μm screen (Collins et al. 2000; Paul et al. 2001b). Results are presented for only the nonhydrolyzable fraction. The acid soluble fraction can be estimated by difference. Nonhydrolyzable C was determined by dry combustion on a LECO, CNS-2000 Elemental Analyzer.

Data were analyzed using the General Linear Model Procedure (SAS Institute, Cary, NC, USA, 1995). Analysis of variance was performed on each parameter measured. Means were separated using protected least square difference. Significance of less than $p<0.05$ was used.

Results and discussion

Soil pH and soil water content

Soil pH ranged from 7.5 in the native sites to 8.3 in the cultivated fields (Table 2). Although pH increased in all sites during the second year of the study, there was no significant difference between cultivated treatments in either year. Semiarid soils are naturally alkaline because limited precipitation and reduced leaching cause carbonates to accumulate and remain in the soil profile (Parker et al. 1983). Calcareous subsoils exposed during land leveling and further mixed with surface soils during cultivation activities may have contributed to the increase in pH. Also, the addition of carbonates present in the ground water and applied during irrigation events can increase soil pH (Lal et al. 1999). Salts present in subsurface irrigation waters of the Columbia Basin are mostly sodium bicarbonates that can increase the alkalinity of soils (Evans et al. 2000). At the time of sampling, water content (Table 2) of 12% was calculated for native and cultivated sites in 2003. In 2004, soil water content at the time of sampling was 9 and 12% for the native and cultivated sites (CA and CB), respectively.

Total organic soil C and N

TOC and TN were similar for the native sites in 2003 and 2004, averaging 4.3 g C kg^{-1} soil and 0.3 g N kg^{-1} soil, respectively (Table 2) Cultivated sites had greater concentrations of TOC than native sites during both years of the study and were significantly different only in the third year after conversion. The increase in TOC in cultivated sites was attributed to C inputs from the incorporation of the native vegetation during land clearing and land leveling operations, crop residues, and compost additions (Tables 3 and 4). Agricultural practices such as crop rotation, incorporation of plant residue and the addition of composts, animal, or green manure were shown to increase SOC and improve soil properties (Rasmussen et al. 1980; Collins et al. 1992; Smith et al. 1993; Rasmussen and Collins 1991). Rotations with high residue producing crops such as corn were shown to increase soil C (Havlin et al. 1990; Zielke and Christenson 1986; Paustian et al. 1997).

Total N values followed a similar trend to TOC with cultivated fields exhibiting higher N concentrations than native sites for both years, but only significantly different in 2004 in the CB series with 3 years of cultivation. The C:N ratio of cultivated fields decreased in the second and third years after conversion and was attributed to the addition of 138 and 276 kg N ha^{-1} from compost in 2002 and 2003, respectively.

In the semiarid shrub-steppe, moisture limits the growth and NPP of native plant communities; reducing inputs of plant residues and limiting the formation of SOM in native soils (Wildung and Garland 1988; Bolton et al. 1990; Smith et al. 1994). Much of the nonliving above ground C of the semiarid shrub-steppe is in the form of standing dead plant shoots with only litterfall and a small portion of roots contributing to SOM. Plant residues remain on the soil surface because of the low annual precipitation and high temperatures are slow to decompose. Therefore, they would

Table 2 Soil pH, water content, and total organic C and N from the surface soil of the native shrub-steppe and cultivated soils sampled from March 2003 and 2004

Site	pH		Water content[a] (%)		Total OC (mg/kg)		Total N (mg/kg)		C:N ratio	
	2003	2004	2003	2004	2003	2004	2003	2004	2003	2004
NS	7.5 b	7.7 a	11	9	4.3 a	4.3 bc	0.30 a	0.31 b	14.6	13.8
CA	7.9 a	8.3 a	12	11	5.0 a	6.0 ab	0.35 a	0.47 ab	14.3	12.8
CB	8.1 a	8.2 a	12	12	4.1 a	6.4 a	0.28 a	0.50 a	14.5	12.7

Values within a column followed by the same letter are not significantly different at $p=0.05$
NS Native sites, *CA* cultivated sites ages 1 and 2 years, and *CB* cultivated sites ages 2 and 3 years at time of sampling
[a]Water content at time of sampling

Table 3 C inputs from the incorporation of the native vegetation

Vegetation	Standing vegetation Mg ha^{-1}	Annual residue input	Annual residue C input Mg C ha^{-1}	Compost input	Total C±input
Native vegetation[a]					
Sagebrush	1.3	0.2	0.1	–	–
Grasses	3.6	0.7	0.3	–	–
Roots	8.8	0.4	0.2	–	3.5
Total	13.7	1.2	0.6	–	3.5±

Plant residues were assumed to be 40% C on a dry weight basis. ± Total C input of sagebrush and grasses is after disturbance; above ground biomass was removed by burning; soil inputs of C are from root material

[a]Above and below ground biomass for a native sagebrush–bunchgrass site in Eastern WA, USA (Rickard and Vaughan 1988)

contribute only a small amount of C to the SOM. Rickard and Vaughan (1988) reported that the above ground biomass in a relatively undisturbed sagebrush–bunchgrass plant community in Eastern Washington, averaged 490 g m^{-2} with a below ground biomass of 880 g m^{-2} for a total production of 1,370 g m^{-2}. The native plant communities assessed in their study were similar to that found in the current study.

For the first year after conversion (Series CA), C inputs from land clearing, compost additions and sweet corn residues were estimated to be 3.5, 1.4, and 1.8 Mg C ha^{-1}, respectively, for a total input of 6.7 Mg C ha^{-1} (Tables 3 and 4). Because the above ground plant biomass of the native vegetation was removed by burning, before land clearing the primary input of C from the native vegetation was from roots (3.5 Mg C ha^{-1}). van der Krift et al. (2001) and others (Puget and Drinkwater 2001; Allmaras et al. 2004; Zibilske and Materon 2005) found that root decomposition averaged 70% during field and laboratory studies, suggesting that under irrigation conditions 2.5 Mg C ha^{-1} of the C added from roots of the native vegetation could be mineralized in the first season. Acid hydrolysis of the compost showed that 73% of compost C comprised the resistant fraction (Table 5). This fraction is composed of aromatic humics and lignin, which are slow to decompose. During the 130-day laboratory incubation of compost, 4.3% of the total C was mineralized, suggesting that 0.4 Mg C ha^{-1} could be lost during the growing season though decomposition (Table 5). Because corn residues were incorporated late in the fall of 2002 minimal decomposition would have occurred before fields were sampled the prior spring (2003) with little C entering SOM pools until the next growing season. Therefore, net C inputs entering the SOM before sampling would have been 2.0 Mg C ha^{-1} (1.0 Mg C from roots of native vegetation and 1.0 Mg C from compost amendments). Soil C increased 0.7 g C kg^{-1} soil from the native soil after the first year of cropping. Assuming a bulk density of 1.3 Mg soil m^{-3}, the increase in SOM would be 1.8 Mg C ha^{-1}, similar to the 2.0 Mg C ha^{-1} value derived. For the second year of cropping, 3.6 Mg C ha^{-1} was added as compost and pea vines with an increase in soil C of 2.6 Mg C ha^{-1} in the surface (20 cm). Making similar assumptions for year 2, net C inputs would be 3.0 Mg C ha^{-1} including C derived from the previous year's remaining corn residue and compost (1.2 Mg C ha^{-1}) additions. For the third year after conversion, C inputs from compost additions and corn residues were 2.8 and 1.2 Mg C ha^{-1}, respectively. Residual C from the native vegetation would be minimal because most of the root C would have been mineralized. Inclusion of the previous 2 years of compost additions resulted in a soil C content of 6.4 g C kg^{-1} soil, a significant 2.1 g C kg^{-1} soil increase above C in soils of the native sites. Differences between C inputs and soil C increases may result from overestimating C inputs derived from native vegetation that were based on the findings of Rickard and Vaughan (1988).

Table 4 Crop yields and C inputs into cultivated soils over 3 years of cropping since conversion

Cultivated sites	Yield Mg ha^{-1}	Residue input	Residue C input Mg C ha^{-1}	Compost C input	Total C input
Series CA					
2002 (sweet corn)	17.7	4.6	1.8	1.4	3.2
2003 (peas)	7.2	1.9	0.8	2.8	3.6
Series CB					
2001 (sweet corn)	20.2	5.2	2.1	1.4	3.6
2002 (peas)	6.7	1.7	0.7	1.4	2.1
2003 (sweet corn)	11.1	2.9	1.2	2.8	4.0

Above ground sweet corn residue estimates were based upon a sweet corn yield to residue ratio of 1:0.26 (H. Collins, personal communication). Plant residues were assumed to be 40% C on a dry weight basis. Soil inputs of C are from root material

Table 5 Total C, average cumulative C mineralized, and resistant C from native shrub-steppe and cultivated soils and from dairy manure compost applied to field sites

Site/year/vegetation	Total C	Average cumulative C mineralized		Resistant C	C_r/C_T
	C_T g kg^{-1}	CO_2–C mg kg^{-1}	Percent of C_T %	C_r[a] g kg^{-1}	%
Native	4.3 a	522 a	12.1	2.4 a	56
Series CA					
Year 1 (2002 corn residues)	5.0 a	799 b	16.0	2.7 a	54
Year 2 (2003 (pea residues)	6.0 b	795 b	13.3	3.2 b	53
Series CB					
Year 2 (2002 pea residues)	4.1 a	891 b	21.7	2.3 a	56
Year 3 (2003 corn residues)	6.4 b	727 b	11.4	3.4 b	53
Compost	126 c	5,440 c	4.3	92 c	73

Soil incubated for 175 days, compost for 130 days
Values within a column followed by the same letter are not significantly different at p=0.05. C content of compost based on nonashed sample
[a] Carbon remaining after a 24 h acid hydrolysis (6 N HCl)

C mineralization

Cumulative CO_2 evolved over 175-day laboratory incubations for soils sampled in 2003 and 2004 showed generally higher cumulative C mineralization in cultivated fields compared to native sites (Fig. 2). The CB series of fields showed a slightly higher cumulative C mineralization than CA sites in 2003 with similar cumulative CO_2 evolved for CB and CA in 2004. Cumulative CO_2–C for 2003 ranged from 562 mg kg^{-1} soil in NS samples to a high 891 mg CO_2–C kg^{-1} soil in cultivated fields. Cumulative CO_2–C values for 2004 were slightly lower ranging from 482 to 727 mg kg^{-1} soil for the native and cultivated sites, respectively. The percent of total C mineralized from the cultivated fields significantly increased during the first year of cultivation then decreased to below the average native level by the third year after conversion, most likely due to the increased additions of compost that had a high percentage (73%) of resistant C (Table 5). The percent of C mineralized from the compost was 4.3% (Table 5) after 130 days of laboratory incubation (Fig. 3).

The ratio of resistant soil C, determined by acid hydrolysis, to total soil C was 56% for the native sites decreasing to an average of 53% for the cultivated fields (Table 5). The difference in resistant C between native and cultivated sites most likely resulted from the mixing of subsurface soil horizons low in C during land clearing and land leveling operations. Acid hydrolysis dissolves the polysaccharide fraction and most of the nitrogenous compounds and is known to leave behind the aromatic humics (Scharpenseel and Schiffman 1977) and modern plant lignin residues as defined by Paul et al. (2001a). The residue of hydrolysis was shown to be comparable to the radio C-dated resistant soil C pool, which on average is 1,400 years older than the remaining SOC (Paul et al. 1997), and would not be degraded during the laboratory incubations.

Rates of C mineralized during extended laboratory incubations (Fig. 4) for the native and cultivated sites indicate that C mineralized during the early stages of incubation for both native and cultivated fields consisted of

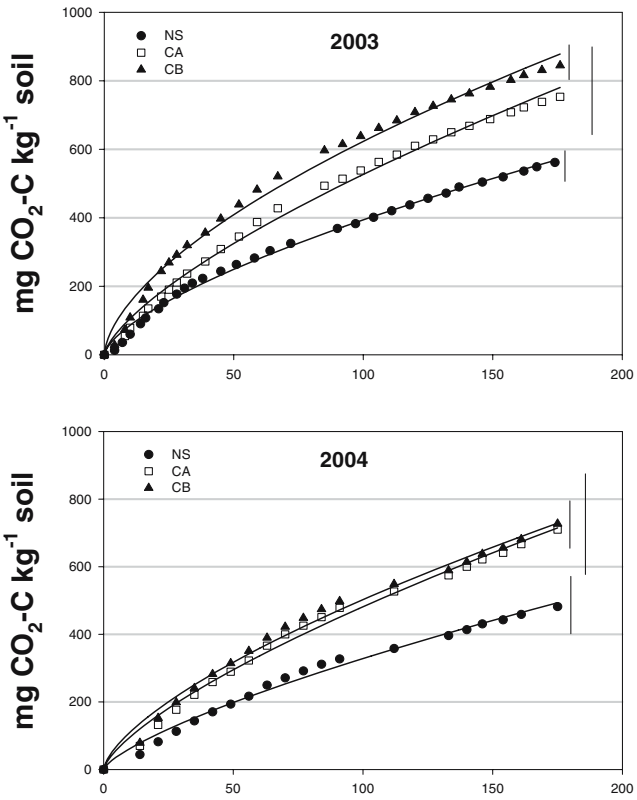

Fig. 2 Cumulative C mineralization curves during 175 days of laboratory incubation for 2003 and 2004 native sites (*NS*), cultivated fields (*CA*) and (*CB*). *Bars* represent standard error at 0.05 for cumulative value of CO_2–C only

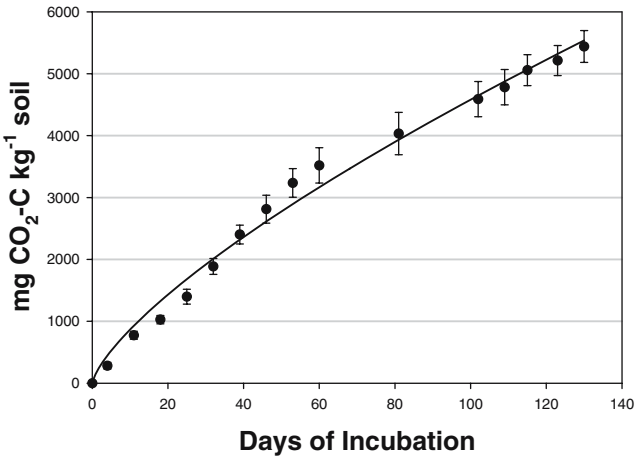

Fig. 3 Cumulative C mineralized from compost amendment incubated at 25°C for 130 days. Compost was applied to cultivated fields before planting

C from the active (C_a) pool. These higher rates reflect variable accumulations of labile C from the incorporation of native and crop residues and compost additions. This pool, on average, contained 4.7% (0.2 g C kg^{-1} soil) and 6.5% (0.5 g C kg^{-1} soil) of the total C in the native sites and cultivated fields, respectively (Table 6). The laboratory MRT of the active pool had significantly shorter turnover times (15–19 days) for cultivated treatments cropped with peas in the second crop year after conversion for both CA and CB series. For the first and third year after conversion, the incorporation of corn residues increased the MRT of the active pool to 25 and 33 days, respectively, with a doubling of the pool above the native sites. Not only did the size of the active pool increase but most likely there were changes in its composition and C:N ratio. A future area for research would be to characterize the chemical composition of the active C pool. Wander et al. (1994) found that soils managed with organic amendments such as animal and green manure and organic composts generally increased active SOM pools.

The proportion of total C in the slow pool (C_s) averaged 44% for the cultivated fields and 40% for the native (Table 6). The increase in the slow pool resulted from the additions of native and crop residues but primarily from the additions of compost. Curve analysis showed that the slow pool increased with increasing years of cultivation. Determination of the size and turnover of the slow pool (C_s) reflected the stabilization of C from compost additions and the effects of management on SOM. Laboratory MRTs of the slow pool ranged from 1.8 years for the native with nearly a 50% increase to 2.6 years by the third year after conversion. By correcting for temperature differences between laboratory and field MAT, field MRTs showed a greater retention of C in the cultivated sites, becoming significantly greater by the third year after conversion. Values for C pools found in this study were consistently lower than those reported in studies conducted in the Great

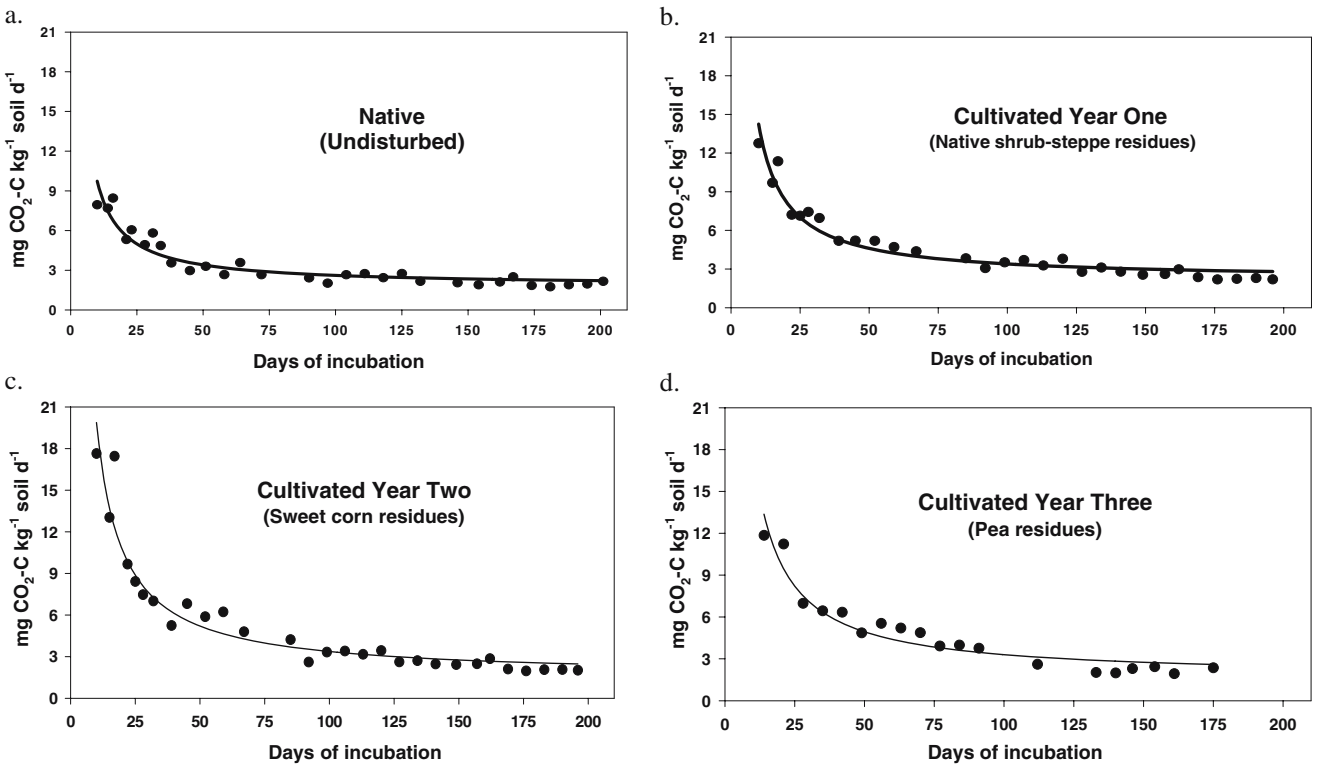

Fig. 4 C mineralization rates for native sites (**a**), cultivated fields for 1 (**b**), 2 (**c**), and 3 (**d**) years after conversion from semiarid shrub steppe to irrigated agriculture

Table 6 Pool sizes and C mineralization kinetics of soil for the active and slow C pools for the native shrub-steppe and cultivated sites by years after conversion

Site/Year/Vegetation	Active pool			Slow pool		
	C_a g kg^{-1}	Lab MRT Days	Field MRT[a]	C_s g kg^{-1}	Lab MRT Years	Field MRT[a]
Native	0.2 a	28 a	77 ab	1.7 a	1.8 a	4.9 a
Cultivated sites series CA						
Year 1 (2002 corn residues)	0.3 a	25 a	68 b	2.2 ab	1.5 a	4.1 a
Year 2 (2003 pea residues)	0.4 b	15 b	41 c	2.4 b	1.8 b	4.9 a
Cultivated sites series CB						
Year 2 (2002 pea residues)	0.5 b	19 b	52 c	2.0 b	1.9 b	5.2 a
Year 3 (2003 corn residues)	0.5 b	33 a	90 a	2.5 b	2.6 b	7.1 b

Values within a column followed by the same letter are not significantly different at p=0.05
[a] MRT or mean residence times converted to field MRTs using a Q_{10} of 2: $\left(2^{(25-t)/10}\right)$ where t is the mean annual temperature (MAT) (t=10.5) (Mummey et al. 1994)

Plains and Midwest Corn Belt (Collins et al. 2000) where total soil organic C ranged from 1.7 (10.7 g C kg^{-1} soil) to 2.8 (18 g C kg^{-1} soil) times more than the TOC found in the third year after conversion. Although Midwestern soils had higher contents of TOC, the distribution of C in C_a, C_s, and C_r fractions was similar. The resistant and slow fractions comprised on average 46 and 50% of the total C, respectively, for the Corn Belt soils. Comparisons of laboratory MRTs for the active fraction were similar, ranging from 26–52 days where MRTs of the slow pool ranged 7–13 years longer in Midwestern soils than those for native and cultivated sites in our study. The greater total concentration of C within the C_s pool was the primary reason for longer MRTs of Midwestern soils.

Conclusion

Conversion from the native shrub-steppe to a managed irrigated agricultural system resulted in increases in pH, TOC, and TN. Rates of C mineralization were greater after conversion with a significant difference between native and cultivated sites. If agricultural land is managed properly, an increase of C sequestration in soils should result. Soil organic C was divided into three pools: an active pool (C_a) consisting of labile C (simple sugars, organic acids, the microbial biomass, and metabolic compounds of incorporated plant residues) with an MRT of days; a slow pool (C_s) consisting of structural plant residues and physically stabilized C, and a resistant fraction (C_r) consisting of lignin and chemically stabilized C. The use of extended laboratory incubations of soil with measurements of CO_2 were widely used to differentiate the C_a and C_s functional C pools in residues (Collins et al. 1990) and soil (Motavalli et al. 1994; Paul et al. 1997). This method constitutes a biological fractionation of SOM where labile fractions are mineralized rapidly by soil biota. C mineralized during the early stages of incubation consisted of C from the C_a pool and reflected variable accumulations of labile C from the incorporation of corn residues. This pool contained 3–5% of the total C and had an average laboratory MRT of approximately 25 days.

The size and turnover rate of the slow pool of C increased with years of cultivation. Positive influences on C storage in this study included crop rotations, addition of organic compost, and residue deposition and incorporation, which were shown to increase SOC over the native shrub-steppe vegetation (Rasmussen et al. 1980; Collins et al. 1992; Smith et al. 1993; Rasmussen and Collins 1991) and water from irrigation that results in increased plant productivity. The system also includes some negative influences on C storage, including harvesting of crops and increased microbial decomposition and volatilization of C as a result of conventional tillage practices and increased moisture.

Improved understanding of the impacts of disturbance from ecosystem conversion and continued cultivation practices may help mitigate negative effects of SOM losses from the emerging agroecosystem.

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